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Response of American dippers (*Cinclus mexicanus*) to variation in stream water quality

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SUMMARY

1. Semi-aquatic birds may be sensitive to altered water quality. While avian species are not used in the bioassessment of streams, they may complement the more common use of benthic macroinvertebrates and fish. We estimated the extent to which water quality can predict attributes of the populations of one common semi-aquatic bird, the American dipper (*Cinclus mexicanus*).
2. First, we estimated dipper presence/absence in relation to water quality as measured by a multimetric assessment index and individual bioassessment metrics. Second, we estimated dipper territory area and reproductive success in response to variation in water quality. We studied the diet, territory area and fecundity of dippers and sampled benthic macroinvertebrates, water chemistry and physical variables at 32 sites with and 17 sites without nesting dippers.
3. Dipper presence was only weakly related to chemical, physical and commonly recorded bioassessment metrics such as per cent Ephemeroptera, Plecoptera and Trichoptera (%EPT). Dippers were strongly related to the abundance of their common prey, *Drunella* and Heptageniidae, which are only a small component of the commonly recorded bioassessment metrics. The variances in territory area and reproductive success were weakly predicted by water quality variables.
4. Dipper presence reflected disturbance as measured by their common prey, showing that lower abundance of these stream invertebrates affected this semi-aquatic bird. We suggest dipper presence/absence might be used in multimetric indices of biotic integrity for the bioassessment of streams.

Keywords: American dippers, benthic macroinvertebrates, bioassessment, sedimentation, terrestrial/aquatic linkages

Introduction

Aquatic animal assemblages respond to stream degradation so predictably that stream ecologists use these assemblages as a tool to assess stream health (Hynes, 1970; Rosenberg & Resh, 1993; Barbour *et al*., 1999). There is also growing knowledge of aquatic to terrestrial linkages (Polis, Anderson & Holt, 1997). Aquatic insect subsidies are known to be important to riparian bird assemblages (Nakano & Murakami, 2001), and understanding how in-stream degradation affects riparian animals may be critical for their conservation and management. We examined a common semi-aquatic bird of Western North America, the American dipper (*Cinclus mexicanus* Swainson) to estimate the extent to which this riparian animal is affected by water quality as measured by commonly recorded bioassessment indices.

Dippers should respond to decreased water quality because they forage primarily on macroinvertebrates and fish (Mitchell, 1968; Ormerod, 1985) in clear-water streams (Price & Bock, 1983; Kingery, 1996). Impacts that reduced the abundances of these food organisms may reduce dipper abundance at polluted sites.

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Pollution-intolerant macroinvertebrates, such as Ephemeroptera, Trichoptera, and Plecoptera, are important food for dippers (Mitchell, 1968; Ormerod & Tyler, 1987); these taxa are especially sensitive to stream pollution such as deposition of fine sediment (McCafferty, 1981; Lemly, 1982). Habitat quality can also lower dipper reproductive success. In Colorado, siltation and food decline was implicated in low dipper reproductive success (Price & Bock, 1983). In Wales (Ormerod & Tyler, 1987) and Scotland (Vickery, 1991), birds at highly acidic sites laid smaller clutches than those at circumneutral sites. Smaller clutch sizes in acidic streams may be caused by female dippers finding insufficient food and/or calcium for breeding (Ormerod *et al.*, 1991). Despite this body of knowledge on dippers, their population dynamics have been rarely related to water quality using bioassessment indices, the tools used in the monitoring and management of streams (but see Edwards, 1991; Sorace *et al.*, 2002). If their dynamics are associated with changes in water quality, then they can be used to help assess water quality and identify streams at risk.

We had two objectives: first, we estimated dipper presence or absence in relation to water quality as measured by benthic macroinvertebrate assemblages, water chemistry and physical variables. We hypothesised that streams with low water quality should have lowered probability of dipper presence. Second, we estimated the degree to which dipper territory area and reproductive success responded to variation in water quality. We hypothesised that dippers on streams with high levels of fine benthic sediments should require larger territories and have fewer fledglings. We quantified the diet, territory area and fecundity of dippers, benthic macroinvertebrate abundance, water chemistry and physical variables at 32 sites with and 17 sites without nesting dippers. We then compared dipper presence, territory area and reproductive success to the abundance of specific taxa in their diet and stream bioassessment indices.

Methods

Study organism

American dippers, *C. mexicanus*, are widely distributed, semi-aquatic birds. Dippers forage under water and eat benthic stream macroinvertebrates and small fish (Ormerod, Boilstone & Tyler, 1985). Dippers are typically found on high slope streams (>6.0% slope) with cascading water (Price & Bock, 1983; Kingery, 1996), although in this study we observed birds nesting on lower gradient streams with 1.0–0.5% slope. Dippers tend to build nests over the fastest moving water in the stream (Feck, 2002). Courtship begins in March to April when birds pair-bond and defend territories; they nest-build in late March to April and females gestate for 14 days; young are nestlings for 2–3 weeks and fledge after 17–21 days. Clutch size varies from three to five eggs and two to four nestlings and fledglings. Dippers may produce a second brood in the same nesting season (Price & Bock, 1983; Kingery, 1996).

Study site selection

We monitored 32 sites with breeding dippers and 17 sites with suitable nesting habitat, but no dippers, within drainages of the Wind River and Wyoming Mountain ranges, Wyoming (Fig. 1). We knew dippers used streams in these ranges based on our previous observations (J. Feck, unpublished data). We had no prior knowledge of the water quality of the streams which we surveyed for dippers and consequently did not use water quality for site selection. Based on several previous dipper studies, we set three criteria by which to select survey sites for dippers: (i) dipper presence during the breeding season depends on the availability of suitable nesting habitat in the form of a bridge, boulder or cliff that abuts the stream (Price & Bock, 1983; Archuleta, 1999; Osborn, 1999); (ii) elevation and slope of sites must be similar to sites where we had prior sightings of dippers and values from other studies (Price & Bock, 1983; Osborn, 1999; Feck, 2002); and (iii) the area must be accessible within 1.5 km distance from roads for dippers to be applicable in rapid bioassessment protocols. Using 7.5-minute United States Geological Survey (USGS) topographical maps, we located all bridges and cliffs on streams draining the Wind River and Wyoming Mountain ranges. We qualitatively ranked the nest site quality from one to four for sites with and without dippers according to previous dipper studies: (i) no ledge under a bridge or cliff present or width of ledge <8 cm; (ii) ledges under a bridge or cliff sloped or narrow, but accessible to predators, not safe from flooding, or inclement weather; (iii) suitable ledges under a bridge or cliff and safe from flooding and

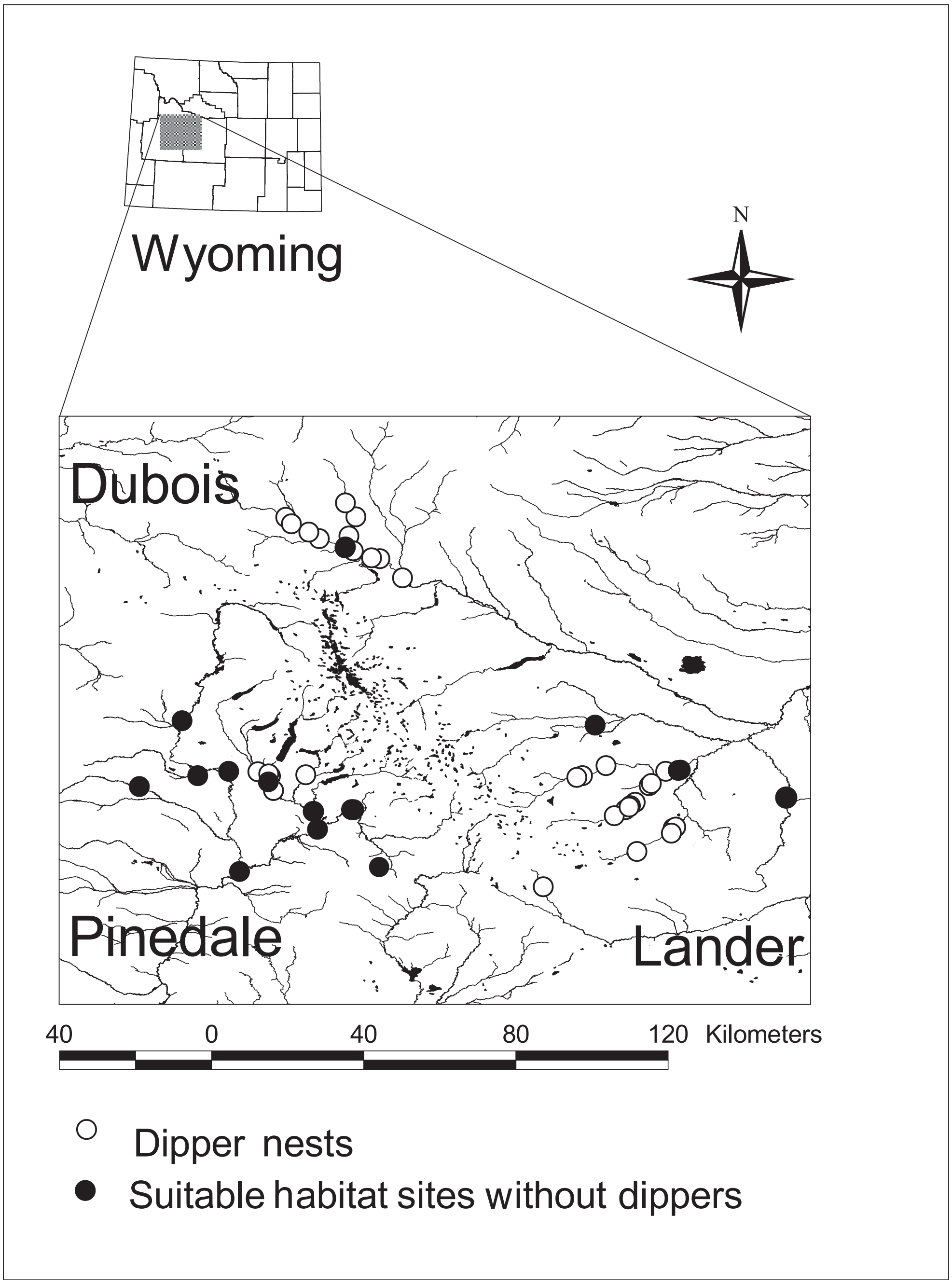

Fig. 1 Map of dipper study sites within the Wind River and Wyoming Mountain ranges, Wyoming, including 32 sites with and 17 sites without dippers.

predators, but no overhanging cover; (iv) if all three criteria met (Price & Bock, 1983; Archuleta, 1999).

During March to July 2000 and 2001, we surveyed all road-accessible stretches of streams around the Wind River Mountains and censused dippers for approximately 300 h by walking streams. Stretches with suitable nest site structures (bridge, cliff or boulder), but no dippers, were revisited one to four times depending on accessibility to census dippers by walking at minimum 400 m upstream or downstream of the structure or along stream reaches with suitable topography. We know of eight pairs of dippers not sampled, and estimate that we sampled approximately 80% of the breeding dippers found in the montane/foothills to plains ecotone in the Wind River Mountains.

Water chemistry and physical habitat

We measured chemical and physical habitat variables once per site during the macroinvertebrate sampling and following the protocols used by the Wyoming Department of Environmental Quality (WDEQ), Water Quality Division (King, 1993). We measured temperature (°C), pH, specific conductivity (μS cm^{-1}), and dissolved oxygen (mg L^{-1}). We estimated sedimentation as the fraction of fine silt cover on the surface of rocks by per cent cover classes (King, 1993): 0–1 (<5%), 1–2 (5–25%), 2–3 (25–50%), 3–4 (50–75%) and 4–5 (>75%) at each of eight 0.085 m quadrats per site using an underwater viewing tube. We estimated elevation from 7.5-minute USGS topographical maps. We measured slope (%) using a clinometer. We measured stream-wetted width at each nest site.

Macroinvertebrate sampling and identification

We collected eight benthic macroinvertebrate samples from riffle/run sections at each of 49 sites using a 330-μm mesh Hess sampler following the WDEQ protocols for streams (King, 1993). All Hess samples were taken from riffles/run sections ca. 30-m long and randomly selected based on distance along the riffle/run and width across the stream. Samples were placed in separate 1-L plastic bottles, preserved in 98% ethanol and sorted in the laboratory. Benthic samples were stained with Phloxine-B to aid in picking, and we only enumerated invertebrates retained on a 500-μm sieve. In the laboratory, invertebrates were separated from sediment and detritus, and were distributed evenly in an enamel tray divided into 30 equal sections. The entire sample was searched for 10 min for 'big and rare' taxa that were picked first to ensure minimal loss of richness in the sample because of the sub-sampling process (Vinson & Hawkins, 1996). We rarified the samples using a fixed-count method by selecting at random 50 individuals per sample for each of eight samples per site, yielding a standard number of 400 invertebrates (Barbour & Gerritsen, 1996; Vinson & Hawkins, 1996) from each of the 49 sites. An analysis of streams in Wyoming showed a 200-invertebrate subsample is the optimal number to subsample considering the cost and statistical power necessary to calculate metrics used in stream bioassessment (Barbour & Gerritsen, 1996), so a 400-organism subsample should be sufficient. We graphed yield effort curves using cumulative taxa of invertebrates for the first 10 sites and found that species richness always levelled off before the eighth sample, demonstrating adequate subsampling effort (Vinson & Hawkins, 1996). Macroinvertebrates were identified to genus with the exception of Chironimidae, Amphipoda and Gastropoda (identified to family), the Hydracarina (identified to suborder), the Oligochaeta (identified to class) and the Nematoda (identified to phylum) using Pennak (1989); Merritt & Cummins (1996) and Stewart & Stark (1993). Voucher specimens were retained for all taxa.

Macroinvertebrate metrics

We calculated metric scores based upon the Wyoming Index of Biotic Integrity (WY IBI), which was developed specifically for bioassessment of Wyoming streams (Stribling, Jessup & Gerritsen, 2000). It is the sum of nine metrics that vary as a function of water quality in Wyoming rivers. These include number of Ephemeroptera taxa, number of Plecoptera taxa, number of Trichoptera taxa, percentage of all individuals that are Ephemeroptera (excluding Baetidae) and Trichoptera (excluding Hydropsychidae), the family-level Hilsenhoff Biotic Index (HBI; Hilsenhoff, 1988), Biotic Condition Index (BCI; Winget & Mangum, 1979), percentage of five dominant taxa, and percentage of scraper. The percentage of five dominant taxa metric is the sum of the percentages of the five most dominant taxa based on the lowest taxonomic resolution. We calculated percentage of scrapers based on the functional feeding group classification of Merritt & Cummins (1996). The sum of the nine metric equations (Table 1) is the WY IBI site score. The metric equations estimate the 95th percentile of a population based on the WDEQ sampled sites, and the 5th percentile for the percentage of five dominant taxa. The WDEQ classifies streams according to seven ecoregions (adapted from Omernik, 1987). All of our sites were within the Middle Rockies West ecoregion, and so are classified according to a range of scores as follows: 'very good' (85–100), 'good' (70–85), 'fair' (45–70), 'poor' (25–45), and 'very poor' (0–25) (Stribling *et al*., 2000).

Table 1 The Wyoming Index of Biotic Integrity (WY IBI) scores stream water quality by ecoregion, and is the sum of nine metric equations ranging from 0 to 100. Below are the nine metric equations for the Middle Rockies West ecoregion (Stribling *et al.*, 2000)

Metric	Equation
Number of Ephemeroptera taxa	$100\frac{\text{EphemTax}}{11}$
Number of Plecoptera taxa	$100\frac{\text{PlecTax}}{8}$
Number of Trichoptera taxa	$100\frac{\text{TrichTax}}{11}$
% Ephemeroptera (excluding Baetidae)	$100\frac{\text{EphemNBPct}}{54}$
% Trichoptera (excluding Hydropsychidae)	$100\frac{\text{TrichNHPct}}{50}$
Hilsenhoff Biotic Index (HBI)	$100\frac{10-\text{HBI}}{8.5}$
% 5 Dominant taxa	$100\frac{95-\text{Dom5Pct}}{45.5}$
% Scraper	$100\frac{\text{ScrapPct}}{54.5}$
Biotic Condition Index (BCI) – uses Community Tolerance Quotient (CTQa)	$100\frac{110-\text{BCI\_CTQa}}{66.5}$

We modified the WY IBI because the WDEQ identifies Chironomidae to genus, Oligochaeta to subclass, and *Baetis* to species. For this reason, our values for the WY IBI are not consistent with the WY IBI calculated by the WDEQ and consequently should be viewed as relative values for the streams used in this study. Therefore, we refer to this metric as the modified WY IBI. Two of the nine metrics were incompatible: percentage of five dominant taxa and the number of Ephemeroptera taxa (B.K. Jessup, Tetra Tech, personal communication). We were able to reconcile the data for the percentage of five dominant taxa metric by recalculating it based on our streams. We ranked the percentage of five dominant taxa values for all sites, then calculated the fifth percentile according to the 49 site locations. We substituted this value into the metric score equation. For the other eight metrics we used the scoring according to the WY IBI.

We also calculated three commonly recorded bio-assessment metrics using the macroinvertebrate data. Ephemeroptera, Plecoptera and Trichoptera (EPT) and %EPT are the taxa richness and relative abundance of the combined insect orders Ephemeroptera, Plecoptera and Trichoptera. We calculated two tolerance indices: the HBI (Hilsenhoff, 1988) and the United States Forest Service Community Tolerance Quotient (CTQa; Winget & Mangum, 1979). The HBI is the weighted average of tolerance values of families of invertebrates from 1 (i.e. taxa found in highly unpolluted water) to 10 (i.e. taxa found at severely polluted water). The CTQa assigns taxa to a tolerance quotient derived from the taxon's tolerance to alkalinity and sulphates, and its selectivity for or against fine substrate size and low stream gradients (Winget & Mangum, 1979). The CTQa is the weighted abundance average of the tolerance quotient and ranges from 2 (i.e. taxa found in highly unpolluted water) to 108 (i.e. taxa found at severely polluted water). In addition, we estimated the taxa richness as the number of taxa per site, and measured abundance as the number of benthic macroinvertebrates per square metre.

Dipper collection and monitoring

From 32 sites we caught dippers using 12-m mist nets placed above bridges or across the stream, and directly from the nest using a 1-m hand net attached to a telescopic pole. We flushed birds into the mist nets by hazing them. Netted dippers were colour banded to estimate the territory boundary. To measure territories, we flushed birds several times to the upper and lower extents of the territory where birds were spotted turning around (Price & Bock, 1983; Archuleta, 1999; Osborn, 1999). We flagged the ends of the territories and marked them on a 7.5-minute USGS topographical map. A Geographic Positioning System was used to point-delineate dipper nests. We calculated the mean distance between the two points as the territory length per pair using ArcView GIS 3.2 (Environmental Systems Research Institute, 1999). Given the high variation in stream widths (3–80 m), we report territory area as length (m) multiplied by average width (m).

We monitored dipper nests a minimum of four times during the breeding season from April to July to count the number of nestlings and fledglings in a brood, and the number of broods per season. We were unable to count the number of eggs per brood because of difficulty in accessing the nest during flood stages. To estimate diet, we forced adult dippers to consume 1.0 mL of 1.5% solution of potassium tartrate causing them to regurgitate (Poulin & Lefebvre, 1995). The emetic was administered orally through a gavageing tube attached to a 1-cc syringe. The bird was detained for 15 min in a dark cooler, and regurgitate and scat were preserved in 95% ethanol. Birds were given a glucose solution and monitored to ensure no obvious adverse effects from the procedure. We did not handle or take diet samples from nestling or fledglings because of inaccessibility to nests and in some instances risk of prematurely fledging the nestlings. No birds died from handling based on repeated surveys of the nests. Diet contents were sorted in the laboratory, where insect parts were removed from regurgitate material and mounted on microscope slides according to genus, family, or order from 10 birds at 10 different sites from 4 streams chosen randomly across the study area. We immersed the insect parts in glycerol, covered with glass slips and sealed with clear nail polish. Insect parts from the gut contents were identified at 40–63× magnification using Pennak (1989); Merritt & Cummins (1996) and Stewart & Stark (1993) and by comparing with whole macroinvertebrates collected at respective sites. We assessed dipper prey items as percentage contribution by number. We searched diet samples for fish vertebrae and bones to assess diet contributions by fishes. We did not search for chaetae to assess diet contributions by Oligochaeta.

Data analysis

We examined the link between dippers and water quality using two approaches: first, we estimated if dipper presence was associated with variation in invertebrate assemblages, water chemistry and physical variables by using logistic regression. Second, we estimated if territory area and reproductive success was associated with variation in invertebrate assemblages, water chemistry and physical variables by using least squares regression. We used dipper gut content data to select *a priori* macroinvertebrates found in large numbers in the diet as predictor variables referred to as common dipper prey (Fig. 2). We calculated abundance and percentages of common dipper prey items including Ephemeroptera, (*Drunella*, Heptageniidae), and Trichoptera taxa. These taxa were used with stepwise linear regression, scatterplot matrices, and Pearson's correlations to select macroinvertebrates, water chemistry and physical variables that most strongly predict dipper presence, territory area and reproductive success.

Given that the gut contents data showed that dippers select certain macroinvertebrates, we used principle components analysis (PCA; Dunteman, 1989) to relate common dipper prey, chemical and physical data with dipper presence. These variables included *Drunella* and Heptageniidae abundance, temperature (°C), pH, specific conductivity (μS cm^{-1}) and fine silt (% cover class).

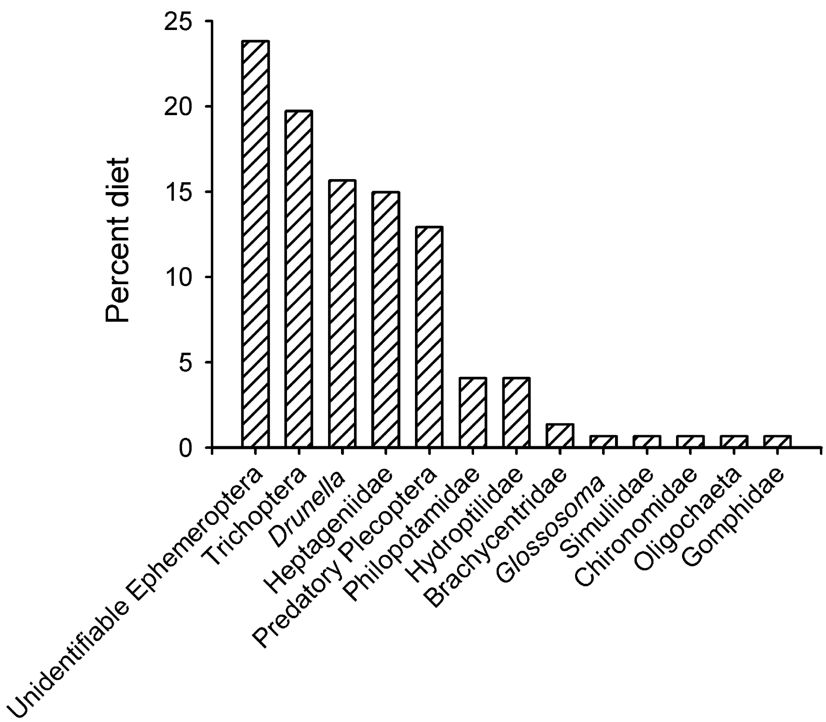

Fig. 2 Per cent contribution to diet based on abundance of diet items from regurgitate and scat samples of 10 dippers from four streams.

We used simple and multiple binary logistic regressions to develop models to predict dipper presence (JMP, 2001). Binary logistic regression allows the calculation of the probability of presence (Hosmer & Lemeshow, 1989):

$$\text{Probability of presence } P = \frac{\exp(\beta_0 + \beta_1)X_i}{1 + \exp(\beta_0 + \beta_1)X_i}$$

where β_0 = the regression intercept, β_1 = regression coefficients, X_i = independent variable.

There are two statistical tests for the significance of the final model in logistic regression. First, we used a likelihood ratio test, with a chi-square distribution, (χ^2_L) that tests the null hypothesis that the excluded parameter is equal to zero. Low *P* values indicate a parameter is not equal to zero (Hosmer & Lemeshow, 1989). Second, we used a goodness of fit test (χ^2_G) that tests the null hypothesis that the data fit the model (Hosmer & Lemeshow, 1989). The goodness of fit test compares the predicted probabilities to the observed probabilities, and higher *P* values indicates better fit of the data to the model (Hair *et al.*, 1998).

We used simple and multiple linear regressions to relate territory area (km^2) and the number of fledglings to water chemistry, physical variables and commonly recorded bioassessment metrics (JMP, 2001). Territory area, common dipper prey abundance and total invertebrate abundance were log-transformed to satisfy normality. All regression analyses are based on significance level set at $\alpha = 0.05$. All statistical analyses were performed using JMP® version 4.0 software (JMP, 2001).

Results

Physical parameters

Elevation ranged from 1500 to 2300 m, slope varied from 0.5 to 6.0% and stream widths varied from 3 to 80 m. Macroinvertebrate assemblage structure was, in part, a function of physical stream condition. The modified WY IBI scores were higher on streams with less fine silt ($P = 0.0098$) and lower temperature ($P = 0.0079$), based on multiple regression analysis ($R^2_{adj} = 0.39$). Fine silt was weakly negatively related to stream slope ($P = 0.0125$) ($r^2 = 0.13$).

Dipper presence/absence

There was no difference between the estimated quality of the potential nest sites with and without dippers ($t = 1.38$, $P = 0.175$, d.f. = 47) suggesting that sites without dippers had equally suitable nest habitat. Dipper presence was not related to elevation ($\chi^2_L = 1.3$, $P = 0.2603$), but was weakly positively related to slope ($\chi^2_L = 5.39$, $P = 0.0202$). Sites with and without dippers separated when analysed with principle components analysis (Fig. 3). The physical and biological variables explained 67% of the variation in dipper presence. Dipper presence was associated with high abundance of *Drunella* and Heptageniidae taxa, and low pH, specific conductivity, fine silt and temperature (Table 2).

Dipper presence was related to several physical variables, but the model fit was weak. Dipper presence was higher on streams with lower temperature (Fig. 4a), fine silt (Fig. 4b) and pH (Fig. 4c). Temperature varied from 7.0 to 25.4 °C, fine silt scores varied from 0.0 to 4.0 and pH varied from 5.8 to 9.3. The same pattern was observed for commonly recorded bioassessment indices. The WY IBI scores varied from 28 to 53, %EPT varied from 15 to 95%, HBI scores varied from 2.74 to 5.28 and CTQa scores varied from 38 to 99. Dipper presence was higher on streams with low CTQa scores, indicating higher fraction of pollution-sensitive taxa on streams with dippers (Fig. 5a). Dipper presence was more likely on streams with higher modified WY IBI scores (Fig. 5b) and %EPT (Fig. 5c). Despite the statistical significance, these indices were only weakly related to dipper presence and had a weak model fit.

Contrary to physical and commonly recorded bioassessment metrics, the abundance of common dipper prey explained substantially more variance in dipper presence. Dipper gut content samples had high abundance of unidentifiable Ephemeroptera, Trichoptera, *Drunella*, and Heptageniidae (Fig. 2). The probability of finding a dipper was higher on streams with greater *Drunella* (Fig. 6a) and Heptageniidae (Fig. 6b) abundances. At low abundances of *Drunella* and Heptageniidae, dippers may have been present or absent; however, at high abundances of these invertebrates, dippers were always present (Fig. 6a,b). Dipper presence was not related to total invertebrate abundance (Fig. 6c).

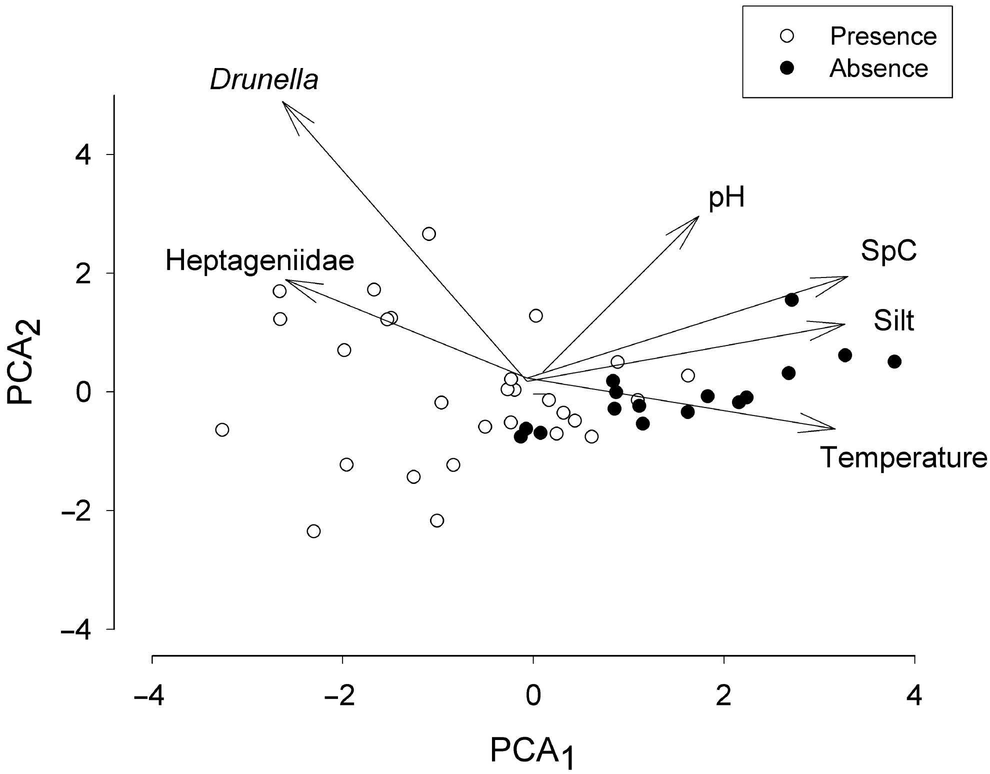

Fig. 3 Principal components analysis axis 1 (PCA_1) and axis 2 (PCA_2) of the combination of chemical, physical and biological variables indicating a separation between sites with (open circles) and without (filled circles) dippers. Arrows indicate direction of factor loadings on axes 1 and 2.

Table 2 Principle component analysis (PCA), eigenvectors for PCA axis 1 and 2 of the abundance of Heptageniidae and *Drunella*, pH, specific conductivity (SpC, μS cm^{-1}), fine silt (% cover class) and temperature (°C)

	PCA axis 1	PCA axis 2
Factor loadings		
Heptageniidae	−0.393	0.287
Drunella	−0.342	0.647
pH	0.314	0.636
Specific conductivity	0.455	0.272
Fine silt	0.449	0.116
Temperature	0.470	−0.089
Eigenvalue	3.05	0.99
Variance explained (%)	50	17
Cumulative variance explained (%)	50	67

The best overall fitted model combined abiotic and biotic variables ($\chi^2_L = 42.3$, $P < 0.0001$, $\chi^2_G = 19.2$, $P = 0.9996$); the abundance of *Drunella* ($\chi^2_L = 31.4$, $P = 0.0000$) was positively related and pH ($\chi^2_L = 15.3$, $P = 0.0001$) was negatively related to dipper presence in a multiple logistic model indicating that dipper presence was higher on streams with greater abundance of *Drunella* and low pH.

Dipper territory area and numbers of fledglings

Territory area decreased as a power function of the density of common dipper prey (percentage of Trichoptera, *Drunella*, and Heptageniidae), (Fig. 7a), showing rivers with greater dipper prey abundance had smaller dipper territories. The exponent of this equation did not differ significantly from −1.0 ($t = 0.03$, $P > 0.1$) indicating territory area decreased directly with the abundance of common dipper prey.

The abundance of common dipper prey explained more of the variance in territory area than did physical and chemical variables, or commonly recorded bioassessment indices. Dippers foraged over smaller areas on streams with greater abundance of per cent Trichoptera, *Drunella* and Heptageniidae (Table 3), lower temperature, pH and specific conductivity (Table 3) and higher modified WY

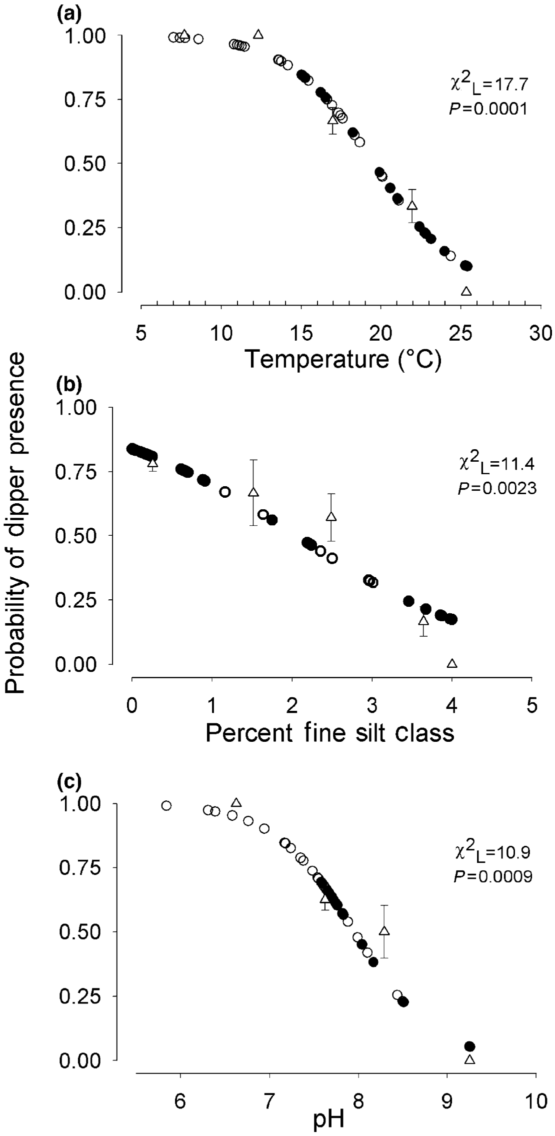

Fig. 4 Logit curves from logistic regression of the probability of dipper presence (open circles) or absence (filled circles) versus (a) temperature [°C; Logit = 7.35 − (0.38 × temperature)] (χ^2_L = 17.7, P = 0.0001; χ^2_G = 38.1, P = 0.6447); (b) fine silt [% cover class; Logit = 1.64 − (0.80 × fine silt)] (χ^2_L = 11.4, P = 0.0023; χ^2_G = 45.6, P = 0.1865); and (c) pH [Logit = 17.47 − (2.20 × pH)] (χ^2_L = 10.9, P = 0.0009; χ^2_G = 37.4, P = 0.4513). The open triangles are observed probabilities with 95% CI at average (a) temperatures within 5° intervals, (b) % fine silt with cover classes 0–1 = <5%, 1–2 = 5–25%, 2–3 = 25–50%, 3–4 = 50–75%, 4–5 = >75%, and (c) pH within one standard unit.

IBI scores (Fig. 7b). Dipper territory area was not predicted by total macroinvertebrate abundance (r^2 = 0.0005, P = 0.904).

The number of fledglings varied slightly across streams (mean = 2.2, SD = 1.0, n = 32), but was

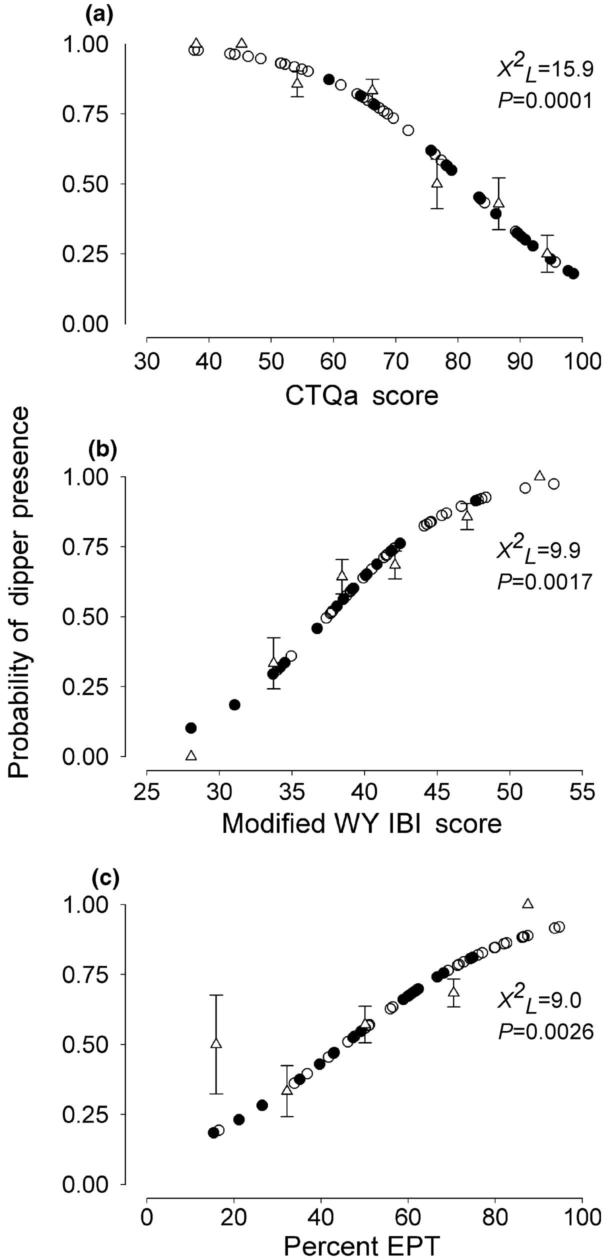

Fig. 5 Logit curves from logistic regression of the probability of dipper presence (open circles) or absence (filled circles) versus commonly recorded bioassessment indices: (a) Community Tolerance Quotient (CTQa) scores [Logit = 7.13 −(0.088 × CTQa)] (χ^2_L = 15.9, P = 0.0001; χ^2_G = 47.4, P = 0.4576); (b) modified Wyoming Index of Biotic Integrity (WY IBI) scores [Logit = −8.69 + (0.232 × $_{\mathrm{mod}}$WY IBI)] (χ^2_L = 9.9, P = 0.0017; χ^2_G = 53.4, P = 0.2416); and (c) per cent Ephemeroptera, Plecoptera and Trichoptera (%EPT) [Logit = −2.247 + (4.95 × %EPT)] (χ^2_L = 9.0, P = 0.0026; χ^2_G = 54.2, P = 0.2184). The open triangles are observed probabilities with 95% CI at average (a) CTQa scores within 10 tolerance quotient intervals, (b) modified WY IBI scores within 5 score intervals, and (c) %EPT within 20% intervals.

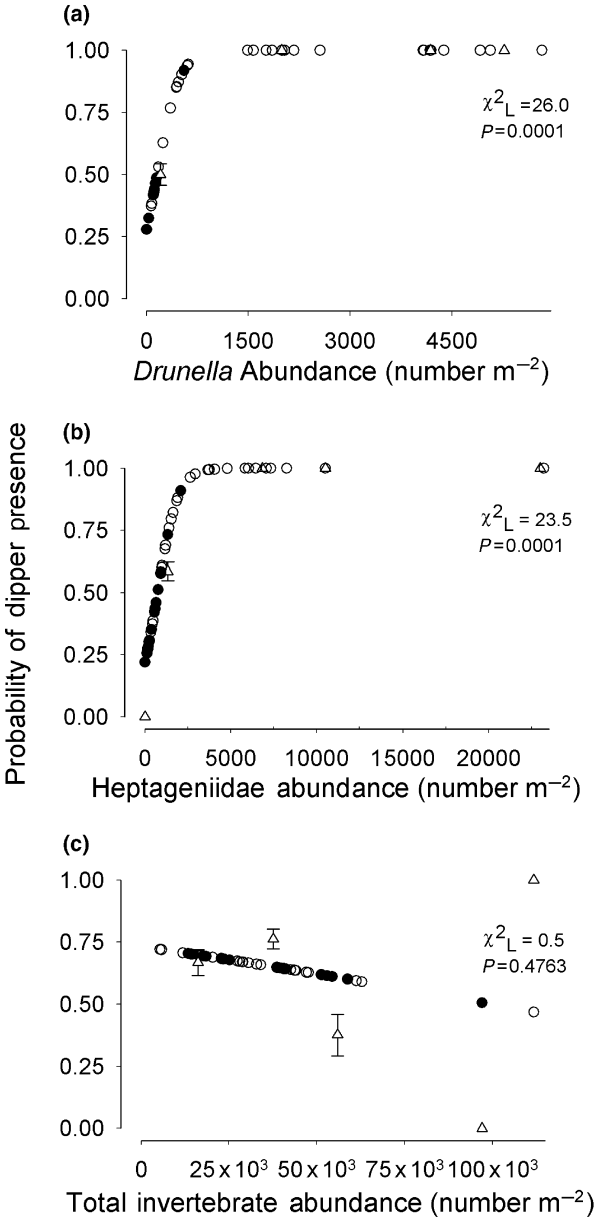

Fig. 6 Logit curves from logistic regression of the probability of dipper presence (open circles) or absence (filled circles) versus invertebrate assemblages: (a) *Drunella* [Logit = −0.95 + (0.006 × *Drunella*)] (χ^2_L = 26.0, P = 0.0001; χ^2_G = 23.2, P = 0.9367); (b) Heptageniidae [Logit = −1.27 + (0.002 × Heptageniidae)] (χ^2_L = 23.5, P = 0.0001; χ^2_G = 39.7, P = 0.6134); and (c) total invertebrate abundance [Logit = 0.997 − (0.00001 × total invertebrate abundance)] (χ^2_L = 0.5, P = 0.4763; χ^2_G = 60.0, P = 0.0668). The open triangles are observed probabilities with 95% CI at average (a) *Drunella* abundance within 1500 (number m^{-2}) intervals, (b) Heptageniidae abundance within 5000 (number m^{-2}) intervals, and (c) total invertebrate abundance within 25 000 (number m^{-2}) intervals.

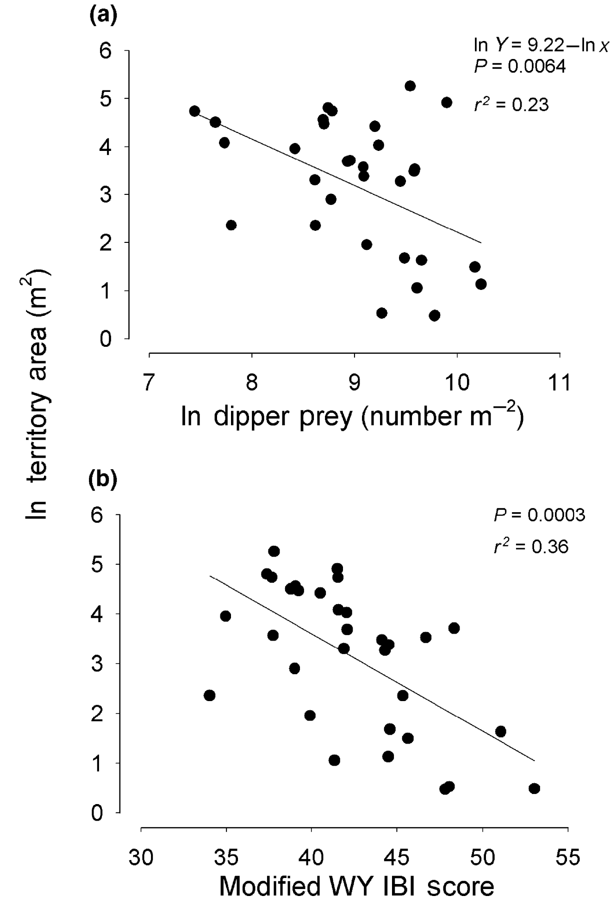

Fig. 7 (a) Territory area (km^2) decreased as a power function of the density of common dipper prey items (number m^{-2}) [ln territory area = 9.22 – ln(common dipper prey)]. (b) Territory area (km^2) was significantly negatively related to the modified WY IBI scores [ln territory area = 11.4 − (0.2 × $_{mod}$WY IBI)]; the line is the least squares estimates of linear regression.

Table 3 Multiple regression, P and R^2_{adj}, of common dipper prey, temperature (°C), pH and specific conductivity (μS cm^{-1}) versus territory area

Variable	Parameter estimate	P	R^2_{adj}
Common Dipper Prey			
Intercept	4.83		
% Trichoptera	−4.97	0.0079	0.57
Drunella	−0.00038	0.0005	
Heptageniidae	−0.00012	0.0042	
Physical/chemical			
Intercept	9.19		
Temperature	0.16	0.0048	0.37
pH	−1.28	0.014	
Specific conductivity	0.0052	0.043	

unrelated to measured biological and physical variables. Dippers fledged more young on streams with higher modified WY IBI scores; however, the regression explained little of the variation in the number of fledglings ($r^2 = 0.11$, $P = 0.07$).

Discussion

Dipper presence/absence and water quality

Dippers responded to decreased water quality. Dippers were more likely to nest on higher slope streams with lower temperature, fine silt and CTQa scores. Stream slope was the weakest predictor despite statistical significance. Stream slope explained little of the variation in fine silt, and is likely a covariate with other variables that are directly related to dippers. CTQa scores (ranging from 38 to 99 of 108) and %EPT (15–95% of 100%) varied substantially, whereas modified WY IBI (28–53 of 100) and HBI scores (2.74–5.28 of 10.0) varied less between sites with and without dippers. The broad range of CTQa scores and variation in fine silt from minimally silted to mostly silted substrate, shows considerable variation in water quality across our study sites. The modified WY IBI and %EPT were even more weakly related to dippers, perhaps because these metrics are too broad with respect to factors that relate to dipper success.

However, abundance of common prey taxa strongly predicted dipper presence. Results from the diet analysis showed that the Ephemeroptera taxa selected by dippers included *Drunella* and Heptageniidae. Although we assessed prey contributions as percentage contribution by number, *Drunella* and Heptageniidae were the second and ninth largest taxa by mass so dominated diet by mass. Furthermore, *Drunella* and Heptageniidae constituted 40% of the Ephemeroptera biomass in our streams and therefore are important prey for dippers. We acknowledge there were other unidentified Ephemeroptera important to dippers, but not all taxa in this order. Thus, using order level measures of Ephemeroptera (e.g. %EPT) across all sites would not discriminate these specific genera selected by dippers.

Common macroinvertebrates in dipper diets was the proximate factor influencing dipper presence. Dippers selected primarily Trichoptera and Ephemeroptera, especially *Drunella* and Heptageniidae. These findings were similar to other studies that found dipper diets during the nesting season contained mostly Trichoptera and Ephemeroptera (Ormerod *et al*., 1985; Ormerod & Tyler, 1991; Tyler & Ormerod, 1992), with Plecoptera constituting only a minor proportion of the diet of adults and nestlings (Ormerod *et al*., 1985). Dipper presence was most closely related to the abundance of *Drunella* and Heptageniidae, and not the total invertebrate abundance, suggesting population sizes of prey found commonly in their diet determined presence. A possible explanation for this pattern is that dippers tend to dive from mid-stream boulders to capture prey on the stream bottom substrate. *Drunella* are large, cobble-dwelling mayflies that most likely remain stationary in the presence of predators (Peckarsky, 1996). Heptageniidae are also cobble-dwelling mayflies that exhibit crawling behaviour in the presence of predators (Peckarsky, 1996). Consequently, both taxa may be easy prey for dippers. Among the identifiable Trichoptera in the diet were Philopotamidae, Hydroptilidae, Brachycentridae and *Glossosoma*. These caddisflies attach to rock surfaces and may be easier prey relative to interstitial taxa.

The best overall model of dipper presence indicated higher probability of dippers on streams with lower pH and higher abundance of *Drunella*. However, the low pH in our streams was substantially higher than in the acidic streams that negatively affected European dipper presence and reproductive success (Ormerod *et al*., 1986; Tyler & Ormerod, 1992). The range of pH at our study sites varied from 6.3 to 8.5. By itself, pH probably did not directly affect dippers, but was most likely correlated with other aspects of water quality that determines dipper presence.

Other studies have shown that lower food supply decreased dipper presence and reproductive success (Price & Bock, 1983; Ormerod *et al*., 1985; Tyler & Ormerod, 1992). However, few studies have documented dipper response to water quality using commonly recorded bioassessment indices. In Great Britain, the density of dippers decreased with organic pollution as measured by the Chandler's Biotic Score. Here low scores occurred on streams with low abundance of Ephemeroptera, Trichoptera and Plecoptera, which are common prey of dippers (Edwards, 1991). More recently, in Italy, the Extended Biotic Index showed the presence of dippers was associated with high quality streams (Sorace *et al*., 2002). Previous studies of the European dipper suggest that low pH negatively affects dipper presence and reproductive success, and their diet is the direct link (Ormerod & Tyler, 1993).

There are few quantitative data linking dipper response to types of pollution other than acidity (Ormerod & Tyler, 1993; but see Edwards, 1991). Although fine sediment is an important pollutant (Waters, 1995), few studies have linked sedimentation to dipper populations (but see Price & Bock, 1983). In the Colorado Front Range, heavy siltation was hypothesised to decrease macroinvertebrate abundances, which in turn reduced the number of dipper fledglings from 21, documented the previous year, to just four young (Price & Bock, 1983). In Wyoming, the primary non-point source pollution affecting streams is sediment (Gumtow, 1992; King, 1993). Large inputs of sediment can reduce benthic macroinvertebrate abundances (Crouse *et al*., 1981; Waters, 1995), and alter macroinvertebrate assemblages from sediment sensitive to insensitive taxa such as Chironomidae and Oligochaeta (Wood & Armitage, 1997). *Drunella* and Heptageniidae are pollution sensitive taxa. The HBI tolerance value for *Drunella* is 0 and Heptageniidae is four of a possible 10 (i.e. low values indicates pollution sensitivity) (Hilsenhoff, 1988). The CTQa tolerance quotients, derived in part from fine sediment, for *Drunella* is 11 and Heptageniidae is 48 out of a possible 108 (low values indicates pollution sensitivity) (Winget & Mangum, 1979), suggesting these taxa are sensitive to sediment. In Idaho, *Drunella doddsi* did not occur in streams with more than 37% fine sediment and were classified as moderately intolerant to fine sediment (Relyea, Minshall & Danehy, 2000). Considering our findings and those of Relyea *et al.* (2000), one possible mechanism for dipper presence is fine sediment controlling dipper prey, or sediment-sensitive invertebrates.

Dipper territory area, reproductive success, and water quality

Water quality variables weakly predicted dipper territory area. Dipper territory area was negatively related to common dipper prey abundance (i.e. Trichoptera, *Drunella* and Heptageniidae). Territory area should relate inversely to the density of available prey (Kodric-Brown & Brown, 1978; Hixon, Carpenter & Paton, 1983), as birds with less abundant prey must forage over larger distances to obtain necessary resources. As this theory predicts, dipper territory area decreased as a power function of the density of common dipper prey.

Other studies of dippers document larger territories at low pH because of decreased prey (Vickery, 1991; Ormerod & Tyler, 1993). The Belted kingfisher (*Megaceryle alcyon*), another stream-obligate avian species, had territories that were inversely related to food abundance during the non-breeding season. However, during the breeding season when nest sites were a limiting factor, territory size was not correlated to food abundance (Davis, 1982).

Water quality was an even weaker predictor of the numbers of fledglings than was territory area. Presumably dippers maximised the number of young fledged regardless of environmental conditions; hence, the number of fledglings did not vary significantly among sites, even on lower water quality streams. This finding was contrary to other studies, which documented lower numbers of fledglings on highly acidic streams (Ormerod & Tyler, 1987; Tyler & Ormerod, 1992; Vickery, 1992). However, low calcium may be the reason European dippers had fewer fledglings on acidic sites. Streams in the Wind River foothills are alkaline and calcium probably did not restrict fledgling growth.

Dippers as predictors of water quality

An effective biological water quality metric should be sensitive to human disturbance and have low variability (Karr *et al*., 1986). Our study suggested dipper absence reflected low densities of *Drunella* and Heptageniidae. However, when dippers were present these taxa may have ranged from high to low. The IBI sums a variety of measurements to assess degradation of aquatic ecosystems (Karr, 1981; Stribling *et al*., 2000). The IBI for fish suggests both physical and biotic variables (turbidity, substrate and prey) affect fish assemblages (Fausch, Karr & Yant, 1984). Avian indicators such as the dipper can potentially be integrated with an IBI. Dippers were weakly related to commonly measured bioassessment indices, but strongly related to the abundance of *Drunella* and Heptageniidae, which are pollution- and sediment-sensitive invertebrates. Dippers may be useful water quality indicators because they are conspicuous and easier to measure than macroinvertebrates, periphyton and fish. Provided sampling is conducted in areas of suitable nesting sites, slope and elevation, we suggest dipper presence/absence be incorporated with the multimetric IBI used in the conservation and management of Western, montane streams.

Of growing importance to biological conservation is the use of indicator taxa as proxy measures of ecosystem health (Landres, Verner & Thomas, 1988; Hilty & Merenlender, 2000). Sudden dipper absence in areas of prior occupancy may signal water quality decline (Ormerod & Tyler, 1993; Archuleta, 1999). Human-constructed bridges across streams are used as nesting sites by dippers and are easily monitored for dippers, enabling their disappearance to signal an alteration in water quality such as macroinvertebrate assemblages. Dippers may be visually observed, and therefore monitoring dippers could be more widely applied than benthic macroinvertebrates.

Conclusions

We demonstrated that variation in water quality, as measured by common dipper prey, affected the presence of this semi-aquatic bird. Few studies document the impact of water quality on riparian vertebrate species, despite the growing knowledge of these linkages (Nakano & Murakami, 2001; Sabo & Power, 2002). The link between streams and aquatic avifauna is important because riparian birds are more conspicuous to people than macroinvertebrate assemblages and hence may provide a clearer demonstration of stream degradation to the public. We suggest dippers, like trophy trout, are focal species, which the general public regards as valuable and desires to protect. Dipper presence depended on the abundance of their pollution sensitive prey. Therefore, their presence may indicate the likelihood of an unpolluted stream. Macroinvertebrate assemblages are a current means of assessing water quality (Barbour *et al*., 1999), but at sites with suitable habitat dippers may complement these indices when used as part of a multimetric or multivariate assessment because they prey on macroinvertebrates that indicate higher water quality.

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